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An estimated 2 billion ha of forests are degraded globally and global change suggests even greater need for forest restoration. Four forest restoration paradigms are identified and discussed: revegetation, ecological restoration, functional restoration, and forest landscape restoration. Restoration is examined in terms of a degraded starting point and an ending point of an idealized natural forest. Global change, climate variability, biotechnology, and synthetic biology pose significant challenges to current restoration paradigms, underscoring the importance of clearly defined goals focused on functional ecosystems. Public debate is needed on acceptable goals; one role for science is to inform and help frame the debate and describe feasibility and probable consequences.

KEYWORDS *reconstruction, rehabilitation, reclamation, novel ecosystems, intervention ecology*

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INTRODUCTION

Forests have served human needs for millennia as a source of food, fiber, and spiritual reflection; today the awareness of society's dependence on forests is as strong as ever. Despite this, the unsustainable uses of forests abound and more than 2 billion ha of forests are degraded globally and in need of restoration (Lindenmayer et al., 2012; Minnemayer, Laestadius, & Sizer, 2011). Emerging global concerns over climate change and loss of biodiversity underscore the importance of forestland cover for addressing efforts to reduce emissions from deforestation, sequester carbon in actively growing forest, and sustain habitat and ecosystem function (Alexander et al., 2011; Benayas, Newton, Diaz, & Bullock, 2009; Thompson, 2012). Forest restoration is needed to reverse degradation and increase forest cover so that the remaining, relatively untouched forests can be conserved (Lamb, Stanturf, & Madsen, 2012). Degradation is driven by many social factors—including macroeconomic, demographic, technological, and governance (Kanninen et al., 2007); the relative importance of drivers varies by social context. Although over 12% of forests globally are legally reserved (Food and Agriculture Organization of the United Nations [FAO], 2010), they are often degraded or threatened by encroachment. Even though there is little forest cover loss in the world (FAO, 2010), deforestation is regionally significant (e.g., Sub-Saharan Africa; Kelatwang & Garzuglia, 2006). The scientific community must accept the importance of restoration of degraded forests and respond with approaches to restoration informed by defensible concepts of what defines a forest (Lund, 1999; Putz & Redford, 2010; Wong, Delang, & Schmidt-Vogt, 2007), the threshold between acceptable disturbance and unacceptable degradation (Sasaki & Putz, 2009), and the way current restoration goals should be altered to accommodate future climates (Harris, Hobbs, Higgs, & Aronson, 2006).

Here we present a broad view of restoration and describe and discuss current paradigms and, importantly, terminology surrounding the concept of restoration. We address the question of "What is Restoration?," as various answers have emerged over time, ranging from passive to active approaches (Bradshaw, 1996, 1997), often adding to confusion over terminology (Lamb et al., 2012; Stanturf, 2005; Stanturf & Madsen, 2002). Moreover, the related terms of deforestation and degradation are similarly ill-defined, but an appropriate definition of what is a forest has been critical to international discussions and negotiations. Agreement on basic terminology and how it is applied is crucial to set baselines for evaluating current forest condition and projecting future trends. For example, definitions based on average height at maturity and minimum stem densities are adequate for assessing gains and losses of forest cover but are insufficient for assessing forest condition and the extent of degradation (Putz & Redford, 2010).

What constitutes successful restoration is defined within a cultural and ecological context that also determines what constitutes degradation (Simula & Mansur, 2011). Underlying both is a view of naturalness that mostly goes unexamined (but see Trigger, Mulcock, Gaynor, & Toussaint, 2008). Two critical questions for framing the discussion are (Lenders, 2006), "What are the starting and ending points? What is the practitioner's perception of the extent of human influence in a forested ecosystem, both in the past and the uncertain future?" Our objective is to provide an overview of forest restoration paradigms that guide myriad and expanding interest in forest restoration by placing these paradigms within the context of attempts to increase sustainability. While our focus is on forests, the concepts we discuss more generally apply to many terrestrial ecosystems. We begin by examining starting and ending points, which necessitates an exploration of the implications for restoration practice of a dynamic, open systems view of forest ecosystems. Four approaches to forest restoration are described and their underlying assumptions explored. We introduce a consistent terminology for restoration as viewed through the lens of the goals of the restoration activity. We finish by briefly looking ahead to challenges for restoration presented by ongoing global change (Harris et al., 2006; Polasky, Carpenter, Folke, & Keeler, 2011) and advances in biotechnology (Jacobs, Dalgleish, & Nelson, 2013), including synthetic biology (Sherkow & Greely, 2013) and a potential transformative response in intervention ecology (Hobbs, 2013; Hobbs, Hallett, Ehrlich, & Mooney, 2011; Kates, Travis, & Wilbanks, 2012).

RESTORATION GOALS

Restoration is often motivated by vague goals (Clewell & Aronson, 2006) that generally fall within the concept of sustainability; for instance: repairing ecosystem functions or other desired attributes (Ciccarese, Mattsson, & Pettenella, 2012), enhancing or enlarging specific ecosystems and habitat for species of concern (Thorpe & Stanley, 2011), or enhancing ecosystem capital, such as biodiversity (Seabrook, Mcalpine, & Bowen, 2011). Although passive restoration, in which land is abandoned from intensive utilization, such as agriculture, and allowed to develop without intervention, dates back millennia (Flinn & Vellend, 2005), modern approaches to restoration are more active. These active approaches include traditional methods, such as afforestation; and emerging approaches, such as assisted migration as an adaptive strategy for climate change (Lunt et al., 2013; Williams & Dumroese, 2013). We begin by examining restoration goals in terms of starting points and ending points, within the context of sociocultural values and ecological understanding. The starting point of our discussion is a degraded forest; the ending point goal is the idealized natural forest.

Starting Point: Degraded Forest

The starting point for restoration is determined by the definition of degraded, because restoration is the reversal of degradation. Deforestation, as an extreme form of degradation, is the removal of forest cover and conversion to another land use (Schoene, Killmann, von Lüpke, & Wilkie, 2007). Deforestation therefore is relatively easy to define and map as long as there is consensus on the definition of forest versus nonforest. Degradation, however, is more difficult to define and even harder to map. Various definitions for degradation have been put forth from different disciplinary perspectives, such as land degradation from earth sciences (Hudson & Alcántara-Ayala, 2006) and more recently loss of carbon stocks and climate change mitigation (Putz & Nasi, 2009). The international forestry community generally views degradation as "Changes within the forest which negatively affect the structure or function of the stand or site, and thereby lower the capacity to supply products and/or services" (FAO, 2001, 2011; Lamb et al., 2012). Degradation is limited by some definitions to conditions resulting from human activity (Penman et al., 2003; International Tropical Timber Organization [ITTO], 2002). Thus,

> A degraded forest is a secondary forest that has lost, through human activities, the structure, function, species composition or productivity normally associated with a natural forest type expected on that site. Hence, a degraded forest delivers a reduced supply of goods and services from the given site and maintains only limited biological diversity. Biological diversity of degraded forests includes many non-tree components, which may dominate in the under-canopy vegetation. (Convention on Biological Diversity [CBD], 2002, p. 154)

The FAO (2001) definition is indeterminate on what causes degradation but the Convention on Biological Diversity view specifically links human activity to degradation (CBD, 2001). From the standpoint of restoration goals, degradation may have many causes and the broad indeterminate definition fits best: diminished capacity to supply goods and services, whatever the causal agent (Lamb et al., 2012).

A complete inventory of degradation indicators and agents would be a long list; indicators could be grouped according to the attributes of forest sustainability that are diminished. Our incomplete list includes forest extent over the landscape, indicated by total area, fragmentation, or canopy cover (Table 1). Within stand indicators include under- or overstocking, loss of species or structural complexity, or encroachment of other land uses into a stand. Forest condition in terms of health and vitality is another sustainability attribute that can be degraded and detected as increased levels of mortality or in terms of more subtle indicators of thinned crowns and reduced growth.

TABLE 1 Indicators and Agents of Degradation That Affect Sustainability of Forests

Sustainability attributes	Degradation indicators	Degradation agents
Forest extent	Loss of area Fragmentation Decreased crown cover Understocking Overstocking Loss of species Decrease in structural complexity Encroachment of nonforest uses	Conversion to nonforest (deforestation) Roads Natural disasters Toxic chemicals (via air, water, soil) Small-scale agriculture Salinization Exploitive harvesting Destructive logging
Forest health and vitality	Crown thinning Reduced growth Increased mortality Increased rot (stem and root) Loss of pollinators Loss of seed dispersers	Conversion to simpler structure Altered fire regime Altered inundation regime Overabundant herbivores Defaunation
Biodiversity	Loss of area designated for habitat conservation Reduced richness Loss of connectivity Increased dispersal barriers Loss of species of concern Reduced genetic diversity within populations Increased invasive species	
Protective functions	Loss of area designated for protective purposes Loss of surface cover Increased soil loss Increased sediment delivery	
Productive functions	Reduced site potential Altered nutrient cycling Reduced stocking (below an acceptable level) Depletion of valuable species Increased growing stock decline and mortality Lack of regeneration Invasive species Increased wildfire risk	

A one-to-one correspondence between degradation indicators and agents is lacking; most agents will affect multiple sustainability attributes and be expressed in several indicators.

Degradation may be expressed in sustainability terms as the decreased capacity for resistance or resilience when disturbed (Millar, Stephenson, & Stephens, 2007; O'Hara & Ramage, 2013); one critical aspect is the ability of trees to regenerate after disturbance. A loss of pollinators, seed dispersers, or both may hamper regeneration and increase vulnerability to large, infrequent disturbances. Degradation may threaten the delivery of specific ecosystem services of forests including biodiversity, carbon sequestration, and protective and productive functions (Table 1).

Each of these indicators must be assessed within their socioecological context, in light of past conditions, stand development trajectories, and ongoing management. For example, decreased crown cover due to sustainable harvesting is not degrading. But selective removal of only the most commercially valuable species without securing adequate regeneration not only reduces crown cover but causes extirpation. Indicators must also be assessed within the context of the landscape; a diverse landscape with a range of canopy cover conditions promotes overall biodiversity (Oliver et al., 2012; Swanson et al., 2010).

The agents of degradation are many; often they interact and may lead to deforestation (Malhi et al., 2008; Murdiyarso, Brockhaus, Sunderlin, & Verchot, 2012). Thus, no one-to-one correspondence exists between degradation agents and indicators (Table 1). The causes of degradation (and deforestation) may be grouped as macroeconomic, demographic, technological, and governance factors (Kanninen et al., 2007) and primacy of a driver can be quite complicated to unravel (e.g., Hansen, Lund, & Treue, 2009). Moreover, degradation can occur in degrees. For example, land may be classed as marginal, fragile, or degraded (Hudson & Alcántara-Ayala, 2006); or ecosystems as degraded, damaged, or destroyed (Society for Ecological Restoration International [SERI], 2004).

Because forest ecosystems are dynamic, disturbance is more the norm than the exception; it is common to many spatial and temporal scales, at all levels of ecological organization (Beatty & Owen, 2005; Turner, 2010). Even though humans are unquestionably the primary cause of most forest degradation, extensive, infrequent, or severe natural disturbances may diminish the capacity of a forest to supply products and services or both (Dale, Lugo, MacMahon, & Pickett, 1998). When viewed from a human perspective, natural disturbance processes are termed natural hazards when they are expected to occur and result in a negative effect (Table 2). When the hazardous threat is realized, it may rise to the level of a natural disaster and restoration may be an appropriate response to repair lost functions and reduce vulnerability to future events (e.g., Stanturf, Goodrick, & Outcalt, 2007). Clearly a gradient exists from stress to natural disaster with intermediate stages of disturbance and degradation; at each stage, both normal (i.e., natural) and anthropogenic agents operate. But what is a “natural” disturbance in an altered landscape? In fire-adapted ecosystems, for example, large fires would often sweep across the landscape and burn with varying intensity causing stand replacement in some places and burning undergrowth in others, leaving behind a mosaic of conditions (e.g., Turner, 2010). Although this can still be seen today in some lightly populated areas such as the boreal forest, in most places there are too many roads and too much suppression activity to allow for truly natural fire regimes (Covington & Moore, 1994; Phillips, Waldrop, Brose, &

TABLE 2 Natural Disturbances That May Become Natural Disasters*

Geophysical	Earthquake	Ground shaking	
		Tsunami	
	Volcano	Eruption	Lava
			Ashfall
			Blast
	Mass Movement (dry)	Rockfall	
		Avalanche	Snow avalanche
			Debris avalanche
		Landslide	Mudslide
			Lahar (volcanic ash slide)
			Debris flow
		Subsidence	Sudden
			Long-lasting
Meteorological	Storm	Tropical storm	Cyclone (hurricane, typhoon)
		Extratropical cyclone	Winter storm, gale
		Local/convective storm	Thunderstorm/lightening
			Tornado
			Orographic storm
			Derecho
			Snowstorm/blizzard
			Sandstorm/dust storm
Hydrological	Flood	General river flood	
		Flash flood	
		Storm surge/coastal flood	
	Mass Movement (wet)	Rockfall	
		Landslide	Debris flow
			Debris avalanche
		Avalanche	Snow avalanche
			Debris avalanche
		Subsidence	Sudden subsidence
			Long-lasting subsidence
Climatological	Extreme temperature	Heat wave	
		Cold wave	Frost
		Extreme winter conditions	Snow loading
			Icing
			Freezing rain
	Drought	Drought	
	Wildfire	Forest fire	Wildfire
			Megafire
		Land fires (grass, shrub, bush)	Wildfire
			Megafire
Biological	Epidemic	Viral	
		Bacterial	
		Fungal	
		Parasitic	
		Prion	
	Infestation	Insect	Grasshopper/locust
			Beetles, etc.
	Invasive	Plant	
		Vertebrates	
		Invertebrates	Insects, earthworms, mollusks

*Adapted from Below, Wirtz, and Guha-Sapir (2009).

Wang, 2012; Veblen, Kitzberger, & Donnegan, 2000). Defining the boundaries between the stages on the degradation gradient is a subjective, social process (Emborg, Walker, & Daniels, 2012) that relies on an individual's valuing of the condition of the forest; the level of degradation is in the eye of the beholder.

A host of environmental stressors and disturbances affect forest ecosystems, including biotic (e.g., exotic and native insects and diseases and invasive plants and animals) and abiotic (e.g., drought, fire, mass movement, hurricanes, tornadoes, and ice storms). A stressor is any of a variety of actors (e.g., herbivory, flooding, low/high temperature, moisture, etc.) that, at certain levels cause stress to organisms. Stress is the resulting effect of a biotic or abiotic agent to which a member of the ecosystem is more or less adapted. A disturbance is any relatively discrete event in time that disrupts ecosystem, community, or population structure, that moves live biomass to dead biomass pools, and that changes resources and substrate availability or the physical environment (Pickett & White, 1985).

A disturbance event exceeds the stressor threshold because of its intensity, frequency, or spatial or temporal scale. Even then, organisms may be adapted to this level of a high intensity disturbance event but when another threshold is exceeded, a severe disturbance may result in degradation. Thresholds may also be exceeded when multiple disturbances—for example, wildfire, insects, and salvage logging—interact to push ecosystem structure and function beyond natural ranges of variation (Lindenmayer, Likens, & Franklin, 2010). Often, such extreme disturbances may result, however, in loss of productivity or other ecosystem services and society regards their effects as degradation or even disaster. Added to these normal stresses are anthropogenic factors such as pollution, development, and fragmentation. Degradation also includes the removal or alteration of natural disturbances. Examples include dams or levees that alter inundation regimes; wildfire suppression that changes fire regimes; and removal of large herbivores and other "ecosystem engineers," such as beaver (Bradshaw, 2005) or conversely, removal of predators that results in increased herbivore levels (Côté, Rooney, Tremblay, Dussault, & Waller, 2004; Tanentzap et al., 2009) that reduce biodiversity.

The Ending Point: A (More) Natural Forest

Just as degradation is defined in terms of social values, the "natural forest" is a social construct. Natural has multiple meanings (Cole & Yung, 2010) depending upon a person's attitude toward humans as being included or excluded from nature (Burke & Mitchell, 2007; Lenders, 2006). Thus to some, natural means a pristine system lacking any effect of humans. This has been a commonly held view in North America where wilderness is equated to ecosystems without human influence, although this view is being reevaluated (Cole, 2000). A related, but distinctly different view allows for minimal human influence but free from intentional human control (Cole, 2000). This is the world said to be inhabited by indigenous peoples, in harmony with nature and thought to lack the technology to cause widespread alteration of the environment, embodied in the concept of the "ecological Indian" (Krech, 1999). Recent studies, however, have challenged this notion and argue that even societies lacking technology for intensive agriculture nevertheless caused profound environmental changes through extensive land use (Ellis et al., 2013; Foley et al., 2005; Steffen, Crutzen, & McNeill, 2007). An often accompanying concept is that ecosystems with minimal human influence are stable or in equilibrium (Perry, 2002) and self-regulating (Middleton, 1999). Thus, restoration to a more natural state under this steady-state view of forest ecosystems means returning to a condition of historical fidelity (Bradshaw, 2005), with similar species composition and structure as before significant human intervention (Higgs, 2003) or at least within the range of historic variability (Keane, Hessburg, Landres, & Swanson, 2009).

We view a gradient of naturalness from the degraded forest to the idealized state of one without any human influence, indigenous or otherwise. In short, we regard naturalness as a continuum, not a binary state (Stanturf & Madsen, 2002). Understanding and defining the various states along the continuum relies on ecological understanding, which has changed over time and now regards forested landscapes as open rather than closed systems, as dynamic rather than steady-state systems (Oliver & O'Hara, 2005). A dynamic vision of the natural forest emerged from the work of the British ecologist Eustace Jones who showed that a "virgin forest" developed under natural disturbance regimes could result in a variety of structures (Bradshaw, Josefsson, Clear, & Peterken, 2010). Since the 1980s, recognition of the role of disturbances in forest ecosystem development and maintenance, including the importance of dead wood and other legacies of the predisturbance ecosystem, has gained appreciable momentum with publication of seminal works on the topic (Beatty & Owen, 2005; Oliver & O'Hara, 2005; Pickett & White, 1985; Sprugel, 1991; Turner, 2010). Nevertheless, these dynamic views of forest ecosystems are less well accepted among the public and some practitioners (Oliver & O'Hara, 2005).

The idealized naturalness endpoint has been called variously the climax, urwald, pristine, or old-growth; this is the forest that develops without human influence and may persist within a landscape mosaic of actively and passively managed (erroneously termed unmanaged) forests. Whether or not this idealized state exists now or within the last 25,000 yr is debatable (Denevan, 1992; Ellis et al., 2013). Certainly old forests with minimal human interference exist that provide examples of more natural conditions than intensively managed or secondary forests (Matuszkiewicz, Kowalska, Kozłowska, Roo-Zielińska, & Solon, 2013). Therefore, the defining characteristics of "naturalness" are the lack of major human interference for all or most of the lifespan of the oldest trees; complex vegetative structures; native species composition; and historical fidelity in terms of disturbance regimes and proportion in the landscape (Hunter, 1996).

Each of these characteristics, however, must be defined in local terms that reflect local values (Frelich & Reich, 2003). For example, how much human influence is allowed? Even forests that ecologists and the public regard as the most natural have legacies of past human influence (Krech, 1999; Ellis et al., 2013). Are the conditions under which the old forest was initiated and developed still operating today (Bradshaw, 2005; Millar & Woolfenden, 1999)? In many places, conditions have changed; for example, many lowland landscapes have been drastically altered and regional and local drainage patterns disrupted or inundation regimes changed by drainage, levees, or dams (e.g., Mississippi River Floodplain; Stanturf et al., 2000; Stanturf, Schoenholtz, Schweitzer, & Shepard, 2001). Furthermore, the notion of native species is somewhat mutable; for example, the post-glacial dispersal of major trees species in northern Europe is still underway (Bradshaw, 2005). Indeed, as Sprugel (1991) asked, "what is 'natural' vegetation in a changing environment?"

We have posited that naturalness is a continuum with degrees of naturalness defined by structure, composition, and function. It is possible to array the many generalized states of forests on this continuum (Figure 1). Thus, the popular conception of the closest forest state to the idealized natural forest is forests that display old forest characteristics, primarily complex structure (Oliver & Larson, 1996; Oliver & O'Hara, 2005; Palik & Engstrom, 1999). Often as structurally complex, naturally regenerated managed forests may seem quite natural, even if they lack key species. Even some managed artificially regenerated forests may be highly functional, such as many beech forests of Western Europe. Many plantation forests of native species have simple structures but on long-rotation may acquire sufficient understory of native plants to resemble old forests, such as the fire-adapted southern pines in the United States (Brockway, Outcalt, Tomczak, & Johnson, 2005; Phillips et al., 2012). Significant departures from these conditions (loss of structure and function) result in degraded forests. Over time, regrowth forests may approach more natural conditions but may lack the composition or structure of managed forests. Native species growing in short rotation plantations that are intensively managed (SRIC in Figure 1) may, despite simple structure, function at a higher level than similar plantations of exotic species.

Beyond structure and species composition is the requirement for "natural" disturbance regimes. One pervasive aspect of human influence has been alteration of disturbance regimes. Efforts to restore historic disturbance regimes include reintroduction of the agent, such as fire (Kuuluvainen, 2002; Moore, Covington, & Fulé, 1999; Phillips et al., 2012) or restoration of river flows and inundation regime (Hughes, del Tánago, & Mountford, 2012). Other methods seek to emulate the effects of historical disturbance regimes

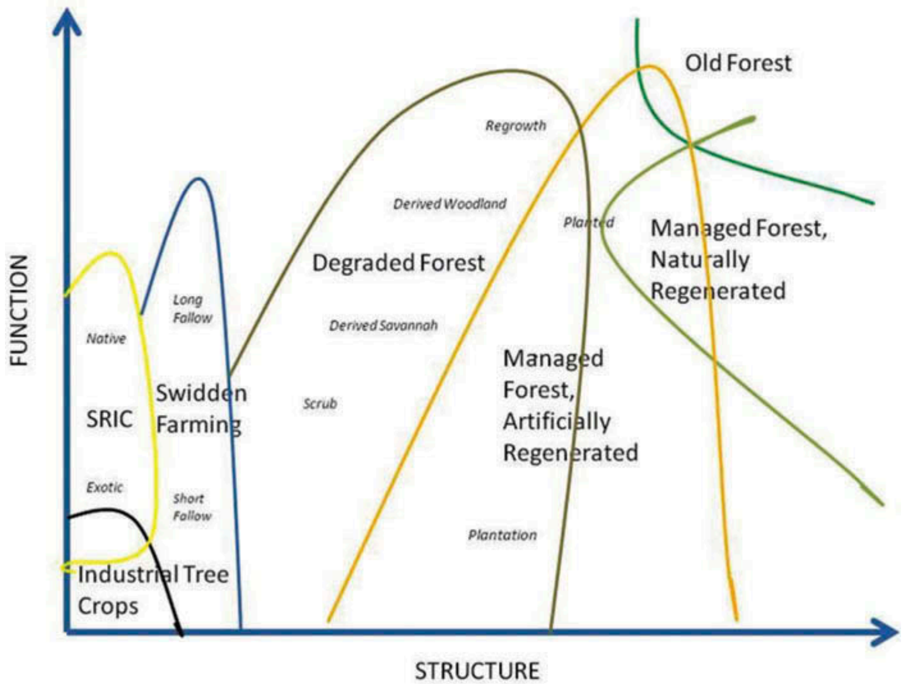

FIGURE 1 Conceptualized forest states in terms of functionality and structure. The degraded to natural continuum is implied, from Industrial Tree Crop to Old Forest. Species composition, particularly native versus nonnative, is contained within functional attributes; for example, short-rotation intensive culture (SRIC) comprised of a native species is higher function compared to SRIC with exotics (adapted from Carle & Holmgren, 2003; Putz & Redford, 2010; Stanturf & Madsen, 2002).

through the manipulation of structure by harvesting (not only removing material but also creating deadwood) or understory composition by planting to emulate the effects of fire or wind; this approach has gained importance in forest management (Laarmann, Korjus, Sims, Kangur, & Stanturf, 2013; Laarmann et al., 2009; Lieffers, Macmillan, MacPherson, Branter, & Stewart, 1996; Lilja, De Chantal, Kuuluvainen, Vanha-Majamaa, & Puttonen, 2005; Long, 2009; Mitchell, Palik, & Hunter, 2002; Seymour, White, & deMaynadier, 2002; Vanha-Majamaa et al., 2007). Emulating natural disturbance regimes may be particularly relevant to restoration of fire-dependent forest (e.g., Franklin & Johnson, 2012; Schwilk et al., 2009) although not all fire surrogate approaches are in-line with mimicking the effects and outcomes of natural disturbance. Rewilding is a related concept to restoration of naturalness from the perspective of conservation biology (Caro, 2007; Donlan et al., 2006; Navarro & Pereira, 2012) or landscape architecture (Convery & Dutson, 2012).

Inherent in the array of potential forest states (Figure 1) is a gradient of degradation extending from the upper right (natural; idealized, predegradation) quadrant to the lower left (highly degraded) quadrant. Transitions from more degraded states to more natural states represent restoration (Lamb, 2011; Stanturf, 2005; Stanturf & Madsen, 2002) whereas transitions in the opposite direction are degradations (Putz & Redford, 2010). To the purist, only a transition to a reference "old forest" end point would constitute restoration; the intermediate transitions in this view are differentiated from restoration and called by other terms, such as rehabilitation (e.g., Simenstad, Reed, & Ford, 2006) or repair (Seabrook et al., 2011). A narrow definition of restoration (as only ecological restoration) excludes, however, much of what is being done in practice (Stanturf & Madsen, 2005; Stanturf, Madsen, & Lamb, 2012) and neglects the conservation importance of earlier successional stages (Donato, Campbell, & Franklin, 2012; Swanson et al., 2010). Nevertheless, an overly broad definition can lead to an "anything goes" version of restoration; although this is likely unacceptable, what is acceptable is a social decision (Brinson, 2000; Emborg et al., 2012).

RESTORATION PARADIGMS

The four current restoration paradigms (revegetation, ecological restoration, forest landscape restoration, and functional restoration) can be differentiated by their goal, or measure of restoration success. Early efforts focused on reducing soil erosion and site degradation; simply planting one or few species was sufficient. At the other end of the spectrum, ecological restoration seeks to restore the vegetation structure and composition as it was prior to disturbance or human manipulation. Two intermediary paradigms, forest landscape restoration and functional restoration, respectively, focused on large-scale restoration and added human needs into the mix and changed the focus from static structural goals to dynamics of ecosystem processes. These paradigms are examined in greater detail below.

Revegetation

Historically, active restoration began as early attempts at restoring ecosystems by focusing primarily on revegetation without much regard for nativeness of species or structural diversity. Sometimes "reclamation" was used to describe efforts to alter natural systems to make them more productive for agriculture or forestry, such as draining wetlands. Early examples of revegetation were primarily aimed at restoring productive functions or avoiding further soil erosion. More modern examples can be found in the efforts to restore degraded farmland or to reclaim mined land. The primary ecological goal of these early restoration programs was revegetation but occasionally a nationalistic or social motivation, such as providing employment, was included (Stanturf, 2005). Examples are restoration of heathland in western Denmark (Madsen, Jense, & Fodgaard, 2005), afforestation in Israel (Orni, 1969), watershed restoration in the southern United States (U.S. Department of Agriculture, Forest Service, 1988), or tree planting by the U.S. Civilian Conservation Corps during the Great Depression of the 1930s to mitigate soil erosion and respond to declining timber resources (Dumroese, Landis, Barnett, & Burch, 2005).

Revegetation has limited value as a contemporary forest restoration paradigm, with the exception of drastically disturbed sites. Simply providing a forest cover, while beneficial in terms of soil protection, may provide few other ecosystem services. Nevertheless, on sites degraded physically, chemically, or both by surface mining, severe soil erosion, or radioactive fallout, for example, the site has been so altered that native vegetation likely will fail. Thus, the revegetation approach fits within the restoration framework as the special case of reclamation and revegetation with exotic species may be a transitory phase (Lamb, Erskine, & Parrotta, 2005; Parrotta, Turnbull, & Jones, 1997).

Ecological Restoration

Ecological restoration, in many ways, is the antithesis of revegetation because its goal is more than simply revegetation, but rather includes specific goals for composition and structure. Ecological restoration is the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed (SERI, 2004). Ecological restoration grew out of practitioner efforts and the realization by primarily academic ecologists that it provided a vehicle to test ecological theory (Bradshaw, 1996). The emerging discipline was advanced by the establishment of the Society for Ecological Restoration and the development of a Primer that provided explicit principles for ecological restoration (SERI, 2004). In the earliest versions of the Primer, the approach was dominated by restoration to past conditions, as exemplified by reference sites. Reference conditions in North America were often the presumed historic conditions before European settlement, which were believed to represent minimal human influence. As noted by Clewell and Aronson (2006), "Descriptions of restoration projects frequently ignore the why of the project and imply that the need for restoration is inherently obvious and its intentions are noble. The underlying reasons to restore remain understated and unappreciated" (p. 421). They offer five rationales (technocratic, pragmatic, biotic, heuristic, and idealistic) for ecological restoration and conclude that none are individually sufficient. The technocratic, pragmatic, and biotic rationales satisfy societal goals; heuristic and idealistic rationales aim for educational and individual goals (Clewell & Aronson, 2006). Insight into how and why ecological restoration projects currently are undertaken is gained from the literature (Burton & Macdonald, 2011) and reports from active restoration projects (Hallett et al., 2013). The relationships among restoration motivations, attributes of restored ecosystems (goals, or definitions of success), and the broad societal goals for restoration identified earlier illustrate the preponderance of repairing ecosystem function, broadly defined (Table 3). Any single project is likely to have multiple goals arising from the motivations of those involved.

Challenges to the notion of one historic past resulted in the concept of multiple possible reference conditions that existed within the range of historical variability (Keane et al., 2009; Landres, Morgan, & Swanson, 1999). Even

TABLE 3 Relationships Among Restored Ecosystem Attributes, Restoration Motivations, and Broader Societal Goals

Restored ecosystem attributes*	Motivations**	Societal goals		
		Enhancing specific ecosystems	Repairing ecosystem function	Enhancing ecosystem capital
Similarity to reference conditions	Promote forest regeneration, diversity		X	
	Culturally important forest types	X		
Presence of indigenous species	General reforestation, afforestation	X		
	Old fields, post-agricultural reclamation	X	X	X
	Protect or establish rare species	X	X	X
Capacity of the physical environment to sustain populations	Reclamation after mining of industrial activity		X	
	Erosion control		X	
	Protect aquatic resources, habitats		X	
	Repair damage from trampling, recreation		X	
Normal functioning	Improve forest health, reduce fire risk		X	
Landscape integration	Habitat connectivity for wildlife		X	
Presence of functional groups			X	
Elimination of threats		X		
Resilience			X	X
Self-sustainability			X	

*Restored ecosystem attributes adapted from Hallett et al. (2013). **Motivations adapted from Burton and Macdonald (2011).

though the requisite condition of minimal human influence was relaxed to accommodate landscapes heavily influenced by millennia of human intervention (such as Europe, Asia, and Africa), an ecological imperative focused on stable ecosystems (Clewell & Aronson, 2013) still dominates the ecological restoration paradigm (Burton & Macdonald, 2011; Hobbs, 2013; Perrow & Davy, 2002; SERI, 2004). In spite of the increasing recognition that restoration to past conditions is generally infeasible, the ecological restoration paradigm remains focused on historical conditions, on stand-level activity, and on "natural" processes and structures (Clewell & Aronson, 2013; Hobbs, 2013).

Forest Landscape Restoration

Forest Landscape Restoration (FLR) differs from site-level restoration because it seeks to restore ecological processes that operate at larger landscape-level scales (Mansourian & Vallauri, 2005). According to Maginnis and Jackson (2007), FLR is defined as "a process that aims to regain ecological integrity and enhance human well-being in a deforested or degraded forest landscape" (p. 10). By this definition, FLR not only broadens the scope of restoration to consideration of the entire landscape but explicitly incorporates human activities and needs. The definition further implies that FLR is a decision-making process and not simply a series of ad hoc treatments that eventually cover large areas (Lamb et al., 2012). One way to look at landscapes is to recognize the biophysical as well as social mosaic of land cover and land use in an area. The variability contained in this landscape mosaic is greater than simply "forest" and "nonforest" (Lindenmayer et al., 2008). Thus, FLR involves choices about how much and where restoration is undertaken, as well as the technical question of how to restore (Palik, Goebel, Kirkman, & West, 2000). Not only must restoration be feasible in terms of conforming to the ecological conditions of particular sites but the restoration techniques used and outcomes desired must meet the socioeconomic constraints of the (often multiple) landowners, land users, or stakeholders (Clement & Junqueira, 2010; Shinneman, Cornett, & Palik, 2010; Shinneman, Palik, & Cornett, 2012).

Finding ways to implement restoration at large or landscape scale is another challenge of FLR (Brudvig, 2011; Frelich & Reich, 2003). This is done by taking a strategic approach; key locations are targeted that return the greatest social benefit (e.g., Maron & Cockfield, 2008; Mercer, 2005; Wilson, Lulow, Burger, & McBride, 2012; Wilson et al., 2011) rather than relying on the individual decisions of separate landholders. Nevertheless, inequities must be avoided and restoration is not carried out at the expense of some landowners or land users, particularly the poorest or least vocal elements of society. A large variety of approaches has been used to address FLR, from relatively informal techniques (Boedhihartono & Sayer, 2012; Palik et al., 2000; Sayer et al., 2013) to computer decision models (Pullar & Lamb, 2012).

Advantages of the forest landscape restoration paradigm over the ecological restoration paradigm include the expanded focus on landscape-level restoration and the explicit inclusion of meeting human livelihoods needs. By recognizing livelihoods and food security needs, FLR is more appropriate in the developing world than ecological restoration, which often has a restore-then-preserve underpinning (e.g., Stanturf et al., 2001). The potential disadvantage of FLR is that it may narrowly focus on current local needs, ignoring broader social needs unless they are included within the mandate of the funding authority. For example, restoring degraded forests with Reducing Emissions from Deforestation and Forest Degradation (REDD) in developing countries, and the role of conservation, sustainable management of forests, and enhancement of forest carbon stocks in developing countries (REDD+) funding would likely take into account local needs because of the need to meet Prior Informed Consent regulations (Costenbader, 2009) but would not necessarily include biodiversity concerns (Alexander et al., 2011).

Functional Restoration

Functional restoration emphasizes the restoration of abiotic and biotic processes in degraded ecosystems. While ecosystem structure and function are closely connected, functional restoration focuses on the underlying processes that may be degraded, regardless of the structural condition of the ecosystem. As such, a functionally restored ecosystem may have different structure and composition than the historical reference condition. King and Hobbs (2006) provide a succinct review on the distinction between functional and structural restoration. They make the point that structural restoration is more focused on a static view of an ecosystem—e.g., historical reference condition, while functional restoration focuses on the dynamic processes that drive structural and compositional patterns. Functional restoration is the manipulation of interactions among process, structure, and composition in a degraded ecosystem. Functional restoration aims to restore functions and improve structures with a long-term goal of restoring interactions between function and structure; for example, reintroducing fire into a longleaf pine forest where long-term fire suppression has allowed a broadleaf midstory to develop (Brockway et al., 2005; Phillips et al., 2012; Schwilk et al., 2009). It may be, however, that a functionally restored system will look quite different than the reference condition in terms of structure and composition and these disparities cannot be easily corrected because some threshold of degradation has been crossed (Whisenant, 2002) or the environmental drivers, such as climate, that influenced structural and (especially) compositional development have changed.

Inclusive Terminology

Terminological distinctions are important and shared understanding facilitates communication. Here it matters whether restoration is seen narrowly (as in ecological restoration) or broadly (as in functional or landscape restoration). Ecological restoration aims to regain the attributes of the historical (i.e., predisturbance) forest (SERI, 2004). We take a broader view of restoration and call this narrow view *Re-creation* and view it as one strategy within the broader panoply of restoration goals.

Other restoration strategies move from a degraded state to a less degraded state (Figure 2); the terminology depends on starting point (Stanturf & Madsen 2002, Stanturf, 2005). Thus, *Reconstruction* is a strategy for restoring forests to lands formerly in other resource land uses, such as agriculture or pasture. Active approaches include afforestation by planting or direct seeding; passive approaches rely on recolonization of open land through natural means such as wind, water, or animal dispersal. Active reconstruction by planting restricted areas, such as is done for stabilizing sand dunes or stream banks or protecting riparian areas with buffer strips, may not be called afforestation but they amount to the same thing—reconstructing a forest where it has been absent for some time and the land has been in other uses. Hybrid approaches also exist, such as overcoming dispersal limitations of

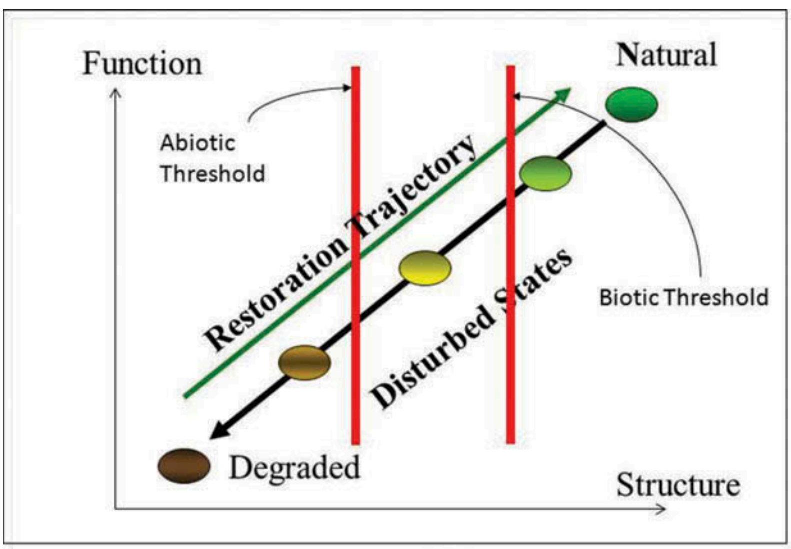

FIGURE 2 The parallel degradation and restoration trajectories in terms of functionality and structure. The intermediate disturbed states (varying degrees of naturalness) are divided by abiotic and biotic thresholds that must be overcome to move to a new stable state. For simplicity these disturbed states are arrayed linearly but in reality, the disturbed ecosystems may be located anywhere and the trajectories can be nonlinear. The Natural endpoint represents an idealized, predisturbance condition (adapted from Bradshaw, 1997).

some species by planting at wide spacing and relying on wind, wildlife, suckering, and other dispersal means to fill the available space with the same or other species (e.g., Scowcroft & Yeh, 2013). In our terminology, afforestation is distinct from reforestation; the latter being the normal forestry practice of establishing a new stand following the removal of the previous stand without an interval of another land use. Some definitions of afforestation set a time criteria (e.g., 25 yr) for the nonforest interlude to accommodate mixed uses such as swidden agriculture (Lund, 1999). Agroforestation is a related strategy to introduce trees onto farms, primarily for carbon sequestration (van Noordwijk, Suyamto, Lusiana, Ekadinata, & Hairiah, 2008).

Rehabilitation in our view is a strategy for restoring desired species composition, structure, or processes, such as fire or flooding regimes to degraded forests. Rehabilitation also refers to removing invasive species. Two specific approaches appear in the literature, conversion and transformation. While these approaches share some characteristics, conversion seems to apply to wholesale removal of an existing overstory and replacement with other species. Transformation applies to a more extended process of partial removals and species replacement but obviously the demarcation between these approaches is fuzzy. For example, if the present condition is a forest lacking a desired structure, selection harvests over time could transform structure with the advantage of maintaining a continuous forest cover (O'Hara, 2001; O'Hara & Ramage, 2013). Alternatively, a conversion approach could call for a clear felling with natural or artificial regeneration (Hansen & Spiecker, 2005). In many cases the availability of markets for removals would determine whether to transform or convert.

Reclamation is a strategy for highly degraded land, such as reclamation of mined land or highly eroded soils (Arnalds, Aradóttir, & Thorsteinsson, 1987; Lamb et al., 2005; Morrison, Lamb, & Hundloe, 2005; Parrotta et al., 1997; Renou-Wilson, Keane, & Farrell, 2008; Renou & Farrell, 2005). Often the restoration site has to be stabilized, mechanically, with temporary vegetation, or both. On such sites it may be necessary to ameliorate soil conditions with amendments to raise pH, add nutrients, and modify bulk density. Thus, the separation between reclamation and rehabilitation may be an abiotic threshold, such as loss of surface soil through erosion or mining (Figure 2). A biotic threshold, such as depauperate species composition, may constitute the threshold between rehabilitation and stand renewal processes; that is, between restoration and sustainable forest management.

Multiple interventions may be required to achieve desired results; time between treatments may be only a few years or it may require decades. For example, reintroduction of fire disturbance may require mechanical or chemical fuel reduction treatments before initiating prescribed burning (Schwilk et al., 2009). Long-term restoration of highly eroded sites may be catalyzed by planting exotic conifers and converting to native broadleaves after one or two rotations (Parrotta et al., 1997). Successful restoration of a landscape may require a combination of strategies with methods tailored to the initial conditions of each site. Sustainable restoration must also anticipate changing conditions and seek to establish robust forest ecosystems that are resilient to future climate shifts.

A CHALLENGING FUTURE AHEAD

Dramatic changes that will result in further forest degradation are likely in global ecosystems (Steffen et al., 2007; Steffen, Grinevald, Crutzen, & McNeill, 2011) and conversely, greater need than ever exists for comprehensive and thoughtful restoration programs. Global changes, from land use changes to feed an ever-increasing world population, to increased international trade that introduces nonnative species to new habitats, to the effects of a generally warmer and drier climate will create additional need to apply restoration techniques. Moreover, the restoration toolbox is expanding through application of biotechnology to existing species and the creation of new species through synthetic biology. The new means to intervene in ecosystem dynamics will intensify debate in scientific circles and among the public about the wisdom of using these new tools. We conclude by briefly examining these challenges.

Global Change

Global ecosystems have been altered by anthropogenic activity to an extent unprecedented in the historic record (Foley et al., 2005; Kareiva, Watts, McDonald, & Boucher, 2007; Zalasiewicz, Williams, Steffen, & Crutzen, 2010). Changes in land cover—such as deforestation and wetland conversion, river channelization and damming, and soil erosion (Sanderson et al., 2002)—are just some of the overt drivers of change leading to loss or diminishment of species, ecosystem functions, and quality of life. Critical changes will affect a variety of ecosystem processes including alteration of the limiting conditions for regeneration and novel pest and disturbance dynamics (Allen, 2009; Ayres & Lombardero, 2000). Native and nonnative species will invade new habitat or change competitive interactions (Bradley et al., 2011; Ricciardi, 2007). Changed conditions will cause effects at variable rates and over a range of scales (Harley & Paine, 2009; Raffa et al., 2008), complicating strategies for responding, especially in regions of mixed land ownerships.

Ecosystems without historical analogs, even in the paleorecord, are now recognized as the norm (Gill, Williams, Jackson, Lininger, & Robinson, 2009; Hobbs, 2013; Hobbs, Higgs, & Harris, 2009). These novel (or emergent, no-analog) ecosystems could result from human intervention, the spread of nonnative species, and climate change. The native versus alien (or natural versus not-natural?) dichotomy has sparked fierce debate in conservation biology and restoration communities (Davis et al., 2011; Hobbs, 2013) with implications for current and future management (Polasky et al., 2011). Issues being debated include moving species beyond their historic range (Ricciardi & Simberloff, 2009) and accepting naturalization of exotic species (Davis et al., 2011).

Climate Change

Changes in climate may degrade forests in two ways. First, in the shorter term, extreme weather events occurring with greater year-to-year variation may degrade existing forests. Climate change is likely to increase weather variability leading to more frequent or increased severity of extreme events (Meehl et al., 2000), especially drought (Allen, 2009; Allen et al., 2010) and severe windstorms. Most climate change research contrasts current conditions with conditions at some future date but is rarely specific on what happens in between; this may implicitly assume a gradual change of mean conditions. One of the salient features of climate change, however, will be more extreme events with greater year-to-year variation in weather (Harris et al., 2006). Such extreme events are predicted to push ecosystems to new (novel) states relatively quickly. Although predictions for large-scale increases in temperature and the intensity and frequency of extreme precipitation and drought are expected, there is less certainty about the location and frequency of extreme weather events at small scales (Van Aalst, 2006). Regardless, predictable changes in climate and unpredictable stochastic natural disturbances resulting from changes in climate will push ecosystems beyond thresholds of acceptable structure and function and thus increase the need for restoration across all scales (Chen, Hill, Ohlemüller, Roy, & Thomas, 2011).

Second, in the longer term, changes are expected in climatic means that may degrade forests as conditions are no longer conducive for their growth (e.g., Lindner et al., 2010; Vose, Peterson, & Patel-Weynand, 2012). This is not unprecedented (Millar & Woolfenden, 1999). Climate was the primary driver of ecological variability and change before significant human impact (mid-19th century). Shifts in climate at various scales constantly change forest compositions, and indeed current forest ecosystems are a response to continually changing climate (Millar & Woolfenden, 1999). Coupled with modeling that suggests the future will also bring no-analog climates (Williams & Jackson, 2007; Williams, Jackson, & Kutzbach, 2007), and because species within a forest ecosystem will respond individually to this radical shift in local climate (Aitken, Yeaman, Holliday, Wang, & Curtis-McLane, 2008), the resulting novel ecosystems will be comprised of species assemblages without current analogs (Harris et al., 2006). These novel ecosystems may be transient with shifts in dominance driven by more variable climate conditions. This assumes, however, that forest plants will be able to move across the landscape in step with changes in climate. Current climate projections require plants, in general, to annually migrate as far as 5 km, almost 10 times faster than their observed rates (Davis & Shaw, 2001). Forest tree species may fall behind (Jump & Penuelas, 2005), thus restoration may necessitate assisted migration (Williams & Dumroese, 2013) based on dynamic transfer guidelines (Potter & Hargrove, 2012) to retain important forest species. This strategy has been criticized (Ricciardi & Simberloff, 2009) mainly because it treats the symptoms of climate change rather than the cause. The stakes (species extinction vs. ecosystem-level degradation) are high (Hewitt et al., 2011), underscoring the need for more dynamic and pragmatic approaches to address the contentious political and ethical questions surrounding assisted migration (Minteer & Collins, 2010).

Biotechnology and Synthetic Biology

Advances in biotechnology and synthetic biology offer the prospect of greater ability to develop new plant material that is better adapted to future climatic conditions, as well as to overcome the loss of keystone species because of introduced pathogens. For example, the American chestnut (*Castenea dentata*) in the eastern United States was practically exterminated by a fungus introduced from Europe (Anagnostakis, 1987). In addition to traditional tree breeding methods using potential resistance within the native population or from the Chinese chestnut, genetic modification of the native species has been successful and transgenic trees may be deployed (Jacobs et al., 2013). Such a strategy might enable the restoration of forests decimated by an extreme weather event by planting genotypes better adapted to warmer, drier conditions (Baker, Diaz, Hargrove, & Hoffman, 2010; Fitzpatrick & Hargrove, 2009; Potter & Hargrove, 2012). New material may include introduced provenances or native plant material modified with genes from related species. Along these same lines, the emerging field of synthetic biology, in which engineering principles are applied to the fundamental components of biology, may lead to “designer organisms” with capabilities unknown in the native population (Rautner, 2001).

The prospect of deploying genetically modified organisms (GMOs), including into native ecosystems, has proven controversial (Strauss & Bradshaw, 2004). Although the public aversion to GMOs is largely visceral, concerns have been raised in forestry over the escape of exotic genes into wild, native populations with unknown consequences and requiring sterility of the GMO is one defense against contaminant genes. Although sterility is acceptable for commercial species that will be artificially regenerated, such as frost-tolerant *Eucalyptus urograndis* (Hinchee et al., 2009), it is a deterrent, however, to the naturalization of a GMO introduced for restoration and sustainable reproduction of a keystone species (Jacobs et al., 2013). An even more radical departure from traditional notions of naturalness arises from the possibility of de-extinction or the cloning of extinct organisms, primarily large Pleistocene mammals (Caro, 2007; Sherkow & Greely, 2013). With the greater ability to manipulate the genetic makeup of trees and the challenges of global change, will there also be a renewed debate about use of native versus nonnative species in restoration?

Intervention Ecology

The challenges of continuing global change and impending climate variability render the goal of restoring to some past conditions even more unachievable (Harris et al., 2006). Recognition that restoration must take place within the context of rapid environmental change has begun to re-define the goals of restoration toward future adaptation rather than a return to historic conditions (Choi, 2007; Choi et al., 2008). This redefinition of restoration removes the underpinning of an ecological imperative (Angermeier, 2000; Burton & Macdonald, 2011) and underscores the importance of clearly defined goals focused on functional ecosystems. Adaptive strategies for coping with climate change may be incremental or transformational (Kates et al., 2012). Transformational adaptations may be responsive or anticipatory, reactive or proactive. Prominent restorationists and conservationists, recognizing the increasing difficulty of returning ecosystems to historic states and the dangers of creating false expectations then failing to deliver, recently have called for a transformational approach to restoration, intervention ecology (Hobbs et al., 2011; Sarr & Puettmann, 2008). Intervention ecology incorporates ecological and socioeconomic aspects and anticipates the need to intervene in governance systems as well (Hobbs et al., 2011), although no specific management goals are implied. Thus, we come back to defining restoration goals that are social choices that should be openly debated in a democratic society. One role for science is to inform and help frame the debate with objective descriptions of feasibility and probable consequences.

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